Appendix 4-7: Updated Ecological Risk Assessment of Mercury at STA-2: Bald Eagle, Wood Stork and Snail Kite

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SUMMARY

This appendix is an update to a risk assessment of methylmercury (MeHg) exposure to birds foraging at Stormwater Treatment Area 2 (STA-2) that was completed in 2004. The previous assessment was required by the Florida Department of Environmental Protection as a special condition to a modification to Everglades Forever Act permit 0126704. This update was requested by the U.S. Fish and Wildlife Service (USFWS) through the U.S. Army Corps of Engineers (USACE) to supplement information provided in the permit application for the expansion of STA-2 and STA-5. Specifically, the USFWS requested (1) an update of the assessment for bald eagles (*Haliaeetus leucocephalus*) foraging at STA-2 to reflect new data on mercury levels in fish collected in 2004; (2) a new assessment for adult wood storks (*Mycteria americana*) foraging at STA-2; and (3) a scoping-level assessment for snail kites (*Rostrhamus sociabilis plumbeus*) foraging at STA-2, using data currently available. In addition to assessing current risk, attempts have been made to extrapolate and forecast future risk, if these receptors were foraging in the expanded area of Cell 4, as currently proposed.

Exposure models were based on literature-derived life history parameters combined with site-specific, measured mercury burdens in fish collected under the permit-mandated monitoring program. Monte Carlo simulations were used to incorporate quantifiable uncertainty and to capture the combination of possible factors leading to worst case exposures. The likelihood that these simulated exposures would be harmful to these birds was then assessed based on the lowest-observed-adverse-effect level and a derived, no-observed-adverse-effect level taken from the peer-reviewed literature.

The results of this risk assessment indicated a low likelihood that MeHg exposures to birds foraging currently at STA-2 would exceed effects thresholds. This places these birds in a low risk category, given the assumptions on which the assessments were based. Furthermore, if current conditions in Cell 3 are a valid reference for predicting future conditions in Cell 4, then future mercury risks to birds foraging in Cell 4 would appear to be negligible. Any remaining uncertainty in the assessment can be offset through a risk management option of monitoring mercury levels in Cell 4 and by having adaptive management strategies in place with explicit decision criteria to respond to undesired, unpredicted outcomes.

INTRODUCTION

This appendix provides supplemental information for a permit application by the South Florida Water Management District (SFWMD or District) for the expansion of Stormwater Treatment Areas 2 and 5 (STA-2 and STA-5) pursuant to a request by the U.S. Fish Wildlife Service (USFWS) through the U.S. Army Corps of Engineers (see correspondence from Oswaldo Collazo, dated May 25, 2005). Specifically, the USFWS requested additional information in the form of ecological risk assessments (ERAs) for the wood stork, bald eagle, and snail kite necessary to concur with the USACE's determination of "may affect, not likely to adversely affect" for these species. Subsequent communications between the District and USFWS (June 17, 2005, teleconference between D. Rumbold, District, and R. Frakes, USFWS) defined the scope of the risk assessments as: (1) an update of a previous assessment for bald eagles foraging at STA-2 to reflect new data on mercury levels in fish collected in 2004, (2) a new assessment for adult wood storks foraging at STA-2, and (3) a scoping-level assessment for snail kites foraging at STA-2 using data currently available.

The document is organized along the major elements of an ERA: problem formulation, exposure analysis, effects analysis, and risk characterization.

PROBLEM FORMULATION

Problem formulation provides the framework for the risk assessment in which ecological receptors are identified and relevant features of the environmental setting, if place-based, are described. This information is combined with an assessment endpoint and an explicit management goal in developing a conceptual model.

In 1994, the Florida legislature enacted the Everglades Forever Act [EFA; Section 373.4592, Florida Statutes (F.S.)] that established long-term water quality goals for the restoration and protection of the Everglades. A crucial element of the plan was the construction of six wetlands, termed stormwater treatment areas (STAs), to reduce phosphorus loading in runoff from the Everglades Agricultural Area (EAA). As such, this action is anticipated to result in beneficial effects in terms of critical habitat restoration.

STA-2 is located in Palm Beach County, Florida (**Figure 1**) immediately west of Water Conservation Area 2A (WCA-2A). The STA is currently comprised of three cells totaling 2,600 hectares that have been in operation since 2000. The eastern and central cells (Cells 1 and 2, respectively) were flooded in 1999 and the western cell (Cell 3) was flooded in 2000. As evident from **Figure 1**, portions of STA-2 were farmed prior to construction. Cell 3 had about 30 percent in sugarcane and 45 percent in sod production. Cell 2 had about 10 percent in sod production (in the northwest corner). In contrast, Cell 1 was not farmed and comprised the Brown's Farm Wildlife Management Area.

The most recent vegetation mapping of STA-2 (Nick Miller, Inc., 2003) designated Cell 3 almost entirely as open water. The northern portion of Cell 3 was dominated by hydrilla (*Hydrilla verticillata*), transitioning to a dominance of pondweed (*Potamogeton illinoensis*) in the south. Cell 2 contained a mixture of sawgrass (*Cladium jamaicense*) and cattail (*Typha* spp.) with occasional patches of duck potato (*Sagittaria lancifolia*) and pickerelweed (*Pontedaria cordata*). The northern portion of Cell 1 contained some open water covered with water lettuce (*Pistia stratiotes*) and monocultures of cattail. Although Cell 2 contained some woody vegetation, Cell 1 was somewhat unique in having a large number of dead oak trees among dense woody vegetation (**Figure 2**).

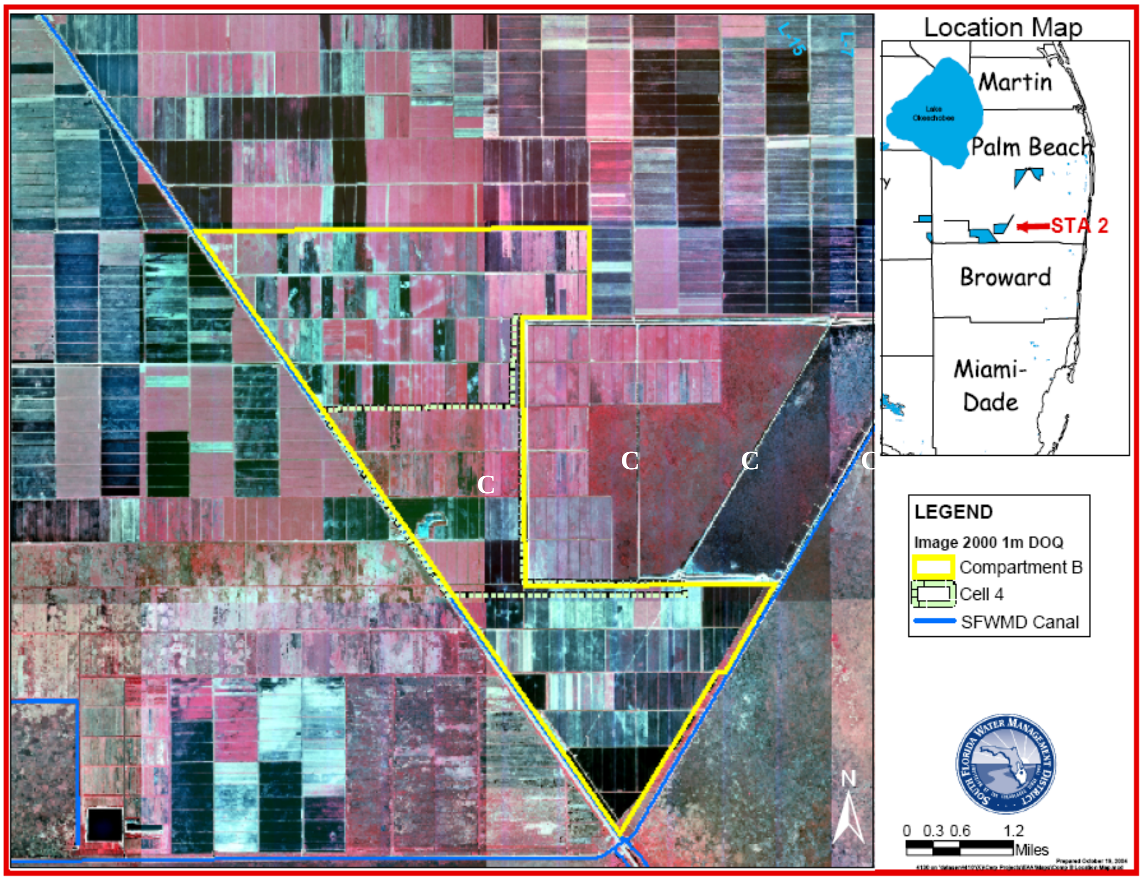


Figure 1. Map of STA-2 showing proposed footprint of Cell 4.

Figure 2. Aerial photograph showing dense, woody vegetation in STA-2, Cell 1 (photo by Kevin Nicholas, SFWMD).

As currently proposed, a fourth cell (i.e., Cell 4) will be constructed immediately west of STA-2 on the former Carroll property (total of 499 hectares) and the former Okeelanta property (316 hectares; Brown and Caldwell, 2005). Like Cells 2 and 3 of the existing STA, the future site of Cell 4 was previously farmed. The Carroll property was used primarily for sod farming; the Okeelanta property was in sugarcane (Brown and Caldwell, 2005). Approximately 1,730 hectares to the north, known as the Woerner South Farm 2 property, are currently being managed as an active sod farm and is the future site of Cell 5. The lease on this land is scheduled to expire in February 2007.

The mercury problem at STA-2 has been detailed elsewhere (Rumbold and Fink, 2002; Fink, 2004; Rumbold and Fink, in press) and will only briefly be summarized here. The Everglades Forever Act permit for STA-2 (No. 0126704) stipulated that flow-through operation of a treatment cell could not begin until the concentrations of total mercury (THg) and methylmercury (MeHg) in the interior marsh were not significantly greater than the corresponding concentrations in the common supply canal. Start-up monitoring of THg and MeHg in surface water began at STA-2 on July 20, 2000. Based on biweekly sampling, Cells 3 and 2 quickly met their mercury start-up criteria in September and November 2000, respectively. In contrast, during this same period, Cell 1 exhibited anomalously high MeHg concentrations in the water column. In response to questions raised by the Florida Department of Environmental Protection (FDEP), the District expanded sampling to include collection of sediment and fish over a 90-day period. The results of the expanded monitoring program were subsequently submitted to FDEP in the form of two reports (Fink, 2002; Rumbold and Fink, 2002). Based on information in those reports, the FDEP granted a modification to the permit in August 2001 to authorize flow-through operation of Cell 1. This modified permit required 12 additional months of expanded monitoring followed by an ERA. Guided by results from the expanded monitoring program, flow rate and water depth were managed in Cell 1 as a means to alter sulfur biogeochemistry and thereby reduce in situ mercury methylation. This adaptive management strategy likely played a role in the decline in water-column concentrations of THg and MeHg in Cell 1 by late 2002 and the subsequent declines in tissue Hg levels in resident fishes. Cell 1 met formal start-up criteria on November 26, 2002. The ERA, which focused on adult and nestling great egrets and bald eagles, was submitted to FDEP (see correspondence to T. Morgan, FDEP, dated March 30, 2004) and subsequently published in a peer-reviewed journal (Rumbold, 2005a).

At the request of USFWS, this report updates the previous assessment for the bald eagle (*Haliaeetus leucocephalus*) to reflect more recent data on mercury levels in fish collected from STA-2 in 2004. Additionally, this assessment will also include two other receptors: the wood stork (*Mycteria americana*) and the Everglade snail kite (*Rostrhamus sociabilis plumbeus*). These three species have special designation under the Endangered Species Act (ESA), as amended (16 U.S.C. 1531, et seq.). The wood stork and Everglades snail kite are designated as endangered; the bald eagle is currently designated as threatened in the conterminous (lower 48) states (Federal Register Vol. 64, No. 128. July 6, 1999; for more information, see the USFWS website at http://www.fws.gov/endangered/). In addition to assessing current risk, attempts will be made to extrapolate and forecast future risk, if these receptors were foraging in Cell 4.

This assessment is based on guidance provided in Guidelines for Ecological Risk Assessment (USEPA, 1998) and Using Probabilistic Analysis in Ecological Risk Assessment (USEPA, 1999).

EXPOSURE ANALYSIS

Potential daily intake by the adult female was estimated using the following simplified equation (adapted from USEPA, 1993):

$$DI_{pot} = \frac{[\sum_{p}(IR \times D_p) \times Hg_P] \times AUf}{BW}$$

Where, DIpot is the potential daily MeHg intake (mg Hg/kg/day), BW is body weight (kg), IR is ingestion rate (kg/day), Dp is dietary fraction of prey species P (%), Hgp is mercury concentration of prey species P (mg Hg/kg), and AUf is an area use factor (unitless weighting factor for receptor use of a given area; discussed in detail in the following section). To incorporate quantifiable uncertainty and to capture the combination of possible factors leading to worst case (i.e., an individual with smaller than average body mass, a diet preference for higher trophic level fish, and ability to capture larger than average, older fish containing higher mercury burdens), a probabilistic approach (i.e., parameters were based on distributions rather than point estimates) was employed through the use of Monte Carlo simulations. Each simulation went through 2000 iterations using a Latin hypercube sampling routine in the Crystal Ball® software (Decisioneering, Denver, CO).

Life history parameter values and distributions used to estimate mercury exposure to the bald eagle are summarized in **Table 1** (for additional details on diet, see Rumbold, 2004). As reported by Rumbold (2004), a bald eagle nest was active in Cell 2 of STA-2 in 2003 (i.e., typically begins in September) and early 2004. Because eagles nested within the STA, the exposure concern was equally for the adult female prior to egg-laying and for the fledgling eaglet. Florida eagles fledge at 11 weeks but remain near the natal nest, the majority of sightings within 0.23 km, until 21 weeks of age (Wood et al., 1998). Accordingly, exposure simulations were carried out for the nestling eaglet, as well as for the adult. In October 2004, the STA was visited and the nest tree was found to have been blown down, most likely as a result of wind damage sustained during one of three hurricanes to hit the region in the summer of 2004. Eagles were not observed nesting or foraging in the STA during the 2004 nesting season or early 2005 (based on field work and monthly helicopter overflights; N. Larson, SFWMD, personal communication, July 7, 2005; note, eagles ware found nesting in the STA in October 2005, after this report was completed). **Figure 3** shows locations of eagle nests near STA-2 that have been reportedly active since 2002. Although foraging distances are not well documented for Florida eagles and distances are expected to vary depending on food availability, STA-2 appears to be outside the range of these active nests (forage distance 3 to 7 km; most active home range < 0.5 km^2, Garrett et al., 1992; USEPA, 1993). Accordingly, the focus of this updated exposure assessment centered on the adult eagle (i.e., exposure to adult female prior to nesting elsewhere).

Life history parameter values and distributions used to estimate mercury exposure to the wood stork are summarized in **Table 2**. The wood stork diet was based primarily on the work of Ogden et al. (1976; **Table 2**); however, distributions were adjusted based on Kushlan (1979), Depkin et al. (1992), Surdick (1998), and Gariboldi et al. (1998), in particular, where adjustments were thought to increase the degree of conservatism in the exposure assessment. For example, largemouth bass were included in the diet, albeit as a minor component, based on a report by Surdick (1998). As another example, distributions for percent biomass of gar and other sunfish

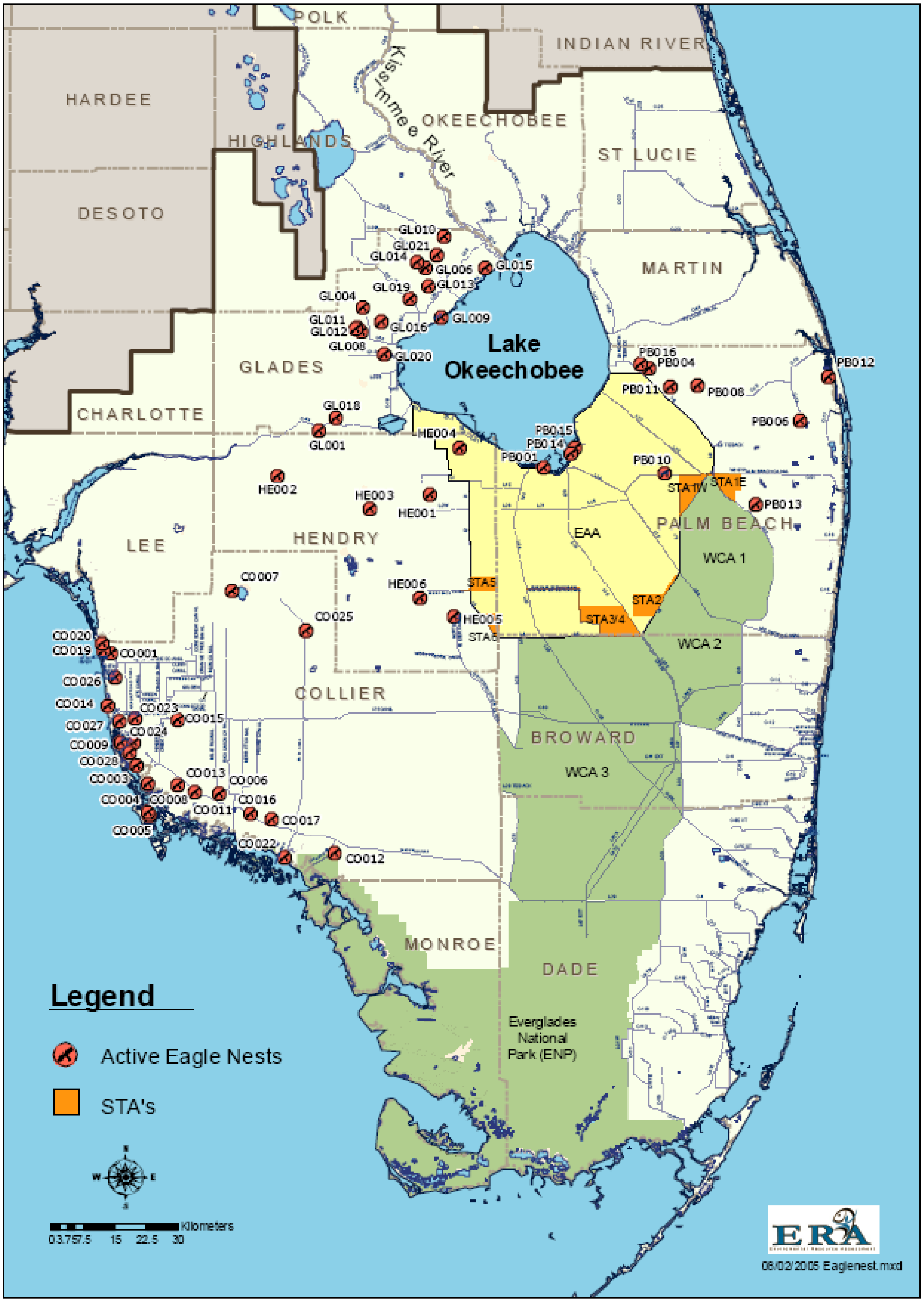


Figure 3. Map showing bald eagle nests that have been reported active in the area surrounding the STAs since 2002 (Florida Fish Wildlife Conservation Commission eagle nest database, http://wildflorida.org/eagle/eaglenests/, July 19, 2005).

Table 1. Life history parameter values and distributions used to estimate mercury exposure to bald eagles.

Life History Parameter	Value (distribution)	Reference
Adult		
Body weight (g)	4500; CV of 4% (normal*)	Wiemeyer, 1991 as cited in USEPA, 1993; USEPA, 1997; CV based on Bortolotti, 1984
Ingestion rate (g d^{-1})	IR = 12% of BW	Stalmaster and Gessaman, 1984 as cited in USEPA, 1993
Diet (% biomass)	Trophic level 4 fish: bass and warmouth: 0, 18, 100 (triangular) Trophic level 3 fish: other Lepomids: 0, 74, 100 (tri) Piscivorous birds: 0, 8, 17	USEPA, 1997; upper range for birds from USEPA, 1993; McEwan, 1977 as cited by USFWS, 1999
Max. prey size (mm)	No size limit	

* Normal distributions were truncated at ± 2 SD to avoid the simulation of implausibly low or high values.

Table 2. Life history parameter values and distributions used to estimate mercury exposure to wood storks.

Life History Parameter	Value (distribution)	Reference
Adult		
Body weight (kg)	2.4±0.35 (normal)	Kahl, 1964; Hancock et al., 1992
Ingestion rate (kg d-1)	IR = $(10^{0.966\log(BW-0.64}/1000)$ BW = bodyweight	Kushlan, 1978; Kahl, 1964
Diet (% biomass)	Fla. gar: 0, 9.2, 42.9[a] Warmouth: 21–38.8, uniform Other Lepomis spp.: 14, 18, 39[b] Largemouth bass: 0–3.2, uniform Y. bullhead: 8–16.4, uniform Small TL 2 & 3 fish: 11–47, uniform	Kahl, 1964; Ogden et al., 1976 Kushlan, 1979; Depkin et al., 1992; Surdick, 1998; Gariboldi et al., 1998
Max. prey size (mm)	≤ 250	Kahl, 1964; Ogden et al., 1976; Surdick, 1998; Gariboldi et al., 1998

[a] Normal distributions were truncated at ± 2 SD to avoid the simulation of implausibly low or high values.
[b] Extremes based on diet in Georgia colonies.

were expanded to include extremes reported for Georgia colonies (Depkin et al., 1992; Gariboldi et al., 1998). Unlike the eagle, which is a raptor and tears apart its prey, wood storks swallow or bolt down their prey whole. Consequently, their prey size is limited by the length and shape of their bill and distensibility of their esophagus. This prey size limitation in storks, egrets, and herons is best illustrated by a report by Ryder (1950) of a dead great blue heron with a 12-inch carp lodged in its gullet. Wood storks are reported to feed almost entirely on fish less than 250 mm (@10 in) in length (Kahl, 1964; Ogden et al., 1976; Depkin et al., 1992; Surdick, 1998; Gariboldi et al., 1998). Surdick (1998) reported that only 3 percent of the stork's prey exceeded 191 mm. However, based on length-to-weight relationships, these few large fish likely represented 58 percent of the biomass ingested. Interestingly, several reports suggest that storks take larger fish in Georgia on average than storks in Florida (M. Coulter, Savannah River Ecology Lab, personal communication, as cited by Hancock et al., 1992; Depkin et al., 1992; for review, see http://www.fws.gov/verobeach/species/birds/wost/wost-msrp/wost_behavior.htm). Wood storks have been reportedly followed 140 km from their colony to feeding sites (see review by Hancock et al., 1992). Obviously, flying long distances (especially via soaring flight) shortens the time available to wood storks for feeding and reduces the number of times an adult stork can return to its nest to feed young (Kahl, 1964). Consequently, during the breeding season, feeding areas close to the colonies may play a larger role (http://www.fws.gov/verobeach/species/birds/wost/wost-msrp/wost_behavior.htm). Most of the food is thought to be caught within 32 km (Kahl, 1964; see **Figure 4**). However, even this may be an overestimation of average flight distances. Bryan and Coulter (1987) report that over 85 percent of the feeding sites were within 20 km of Georgia colonies (mean flight distance of 12.7 ± 10.5 km). Frederick and Collopy (1988) reported a mean distance of 3.4 ± 4.7 km to foraging sites for storks at a Florida colony.

Life history parameter values and distributions used to estimate mercury exposure to the snail kite are summarized in **Table 3**. The diet of the snail kite is almost exclusively apple snails (*Pomacea paludosa*; for review, see Beissinger, 1983; 1990). The apple snail is a primary consumer, feeding on periphyton and macrophyte detritus, and has a relatively short life span (1 to 1.5 yrs). For most of the year, one of two life stages dominate; reproductive adults dominate the population from January until June, at which time these adults undergo rapid senescence and are replaced by young of the year (for review, see Darby et al., 1997). Beissinger (1983) estimated daily energy requirements of the kite at 104.2 kcal per day (cf. 206 kcal and 570 kcal per day for white ibises and wood stork, respectively; Kushlan, 1978; Kahl, 1964).

Table 3. Life history parameter values and distributions used to estimate mercury exposure to snail kites.

Life History Parameter	Value (distribution)	Reference
Adult		
Body weight (kg)	Range 0.35–0.57, mean of 0.446 (normal)	Sykes et al., 1995 as cited by Bennetts and Kitchens, 2000
Ingestion rate (kg d^{-1})	0.150, 0.602, 0.774 (tri)	Beissinger, 1983; W. Kitchens, USGS, pers. comm., July 12, 2005
Diet (% biomass)	Apple snails 100% (point)	Bennetts and Kitchens, 2000

* Normal distributions were truncated at ± 2 SD to avoid the simulation of implausibly low or high values.

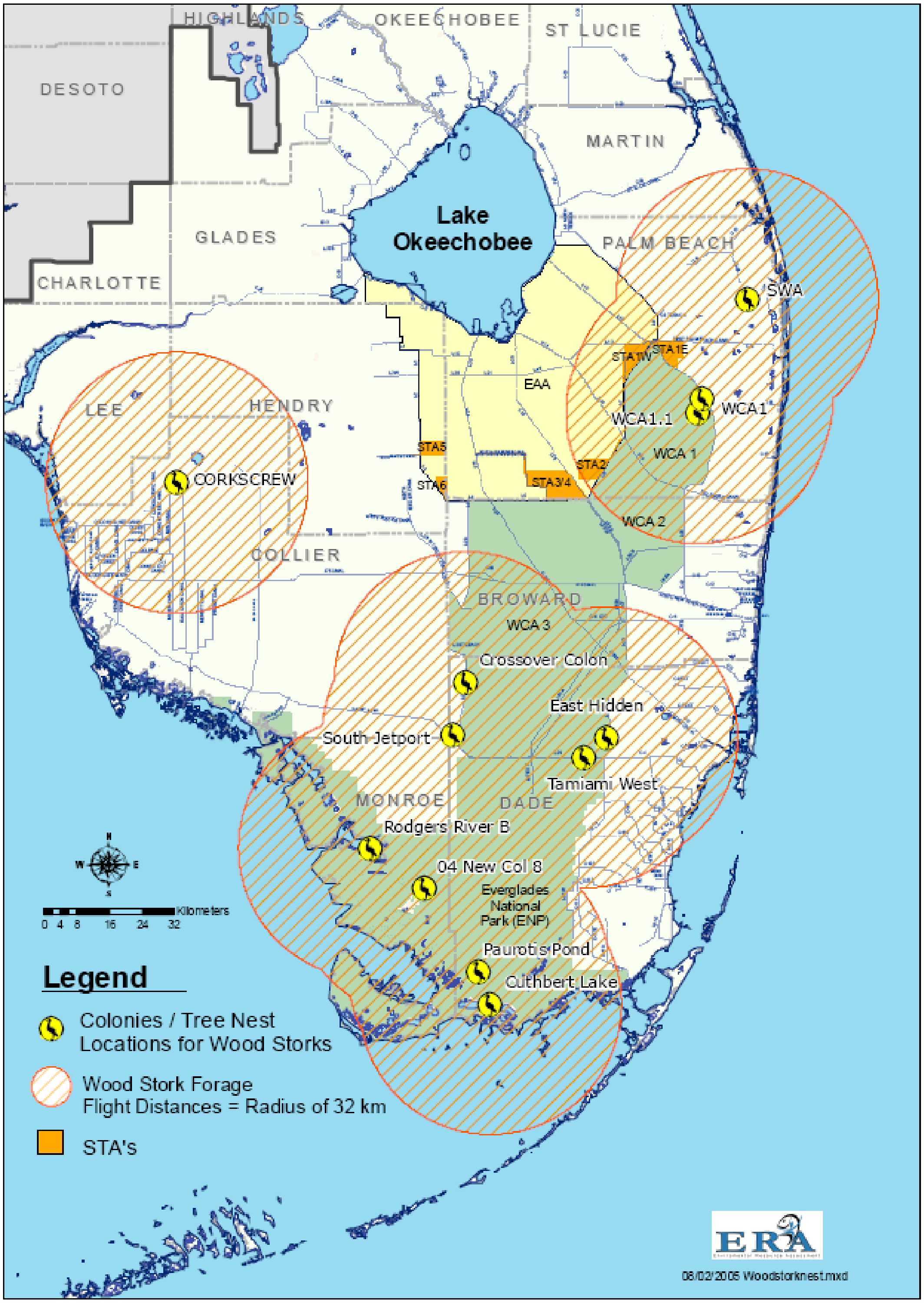


Figure 4. Map showing wood stork nesting colonies that have been reported active in the area surrounding the STAs since 2002 (Gawlik, 2002; Crozier and Gawlik, 2003; Crozier and Cook, 2004).

Based on the caloric value of snails (4.04 kcal per gram dry; Beissinger, 1983), it is estimated that kites require 150.5 grams (wet) of snails per day (approximately 18 snails based on average weight of 8.6 g of 62 snails collected by Eisemann et al., 1997). This represents 33.6 percent of the kite's bodyweight, which is comparable to other published ingestion rates for birds (Welty, 1982; cf. eagles ingest 12 percent BW per day, Stalmaster and Gessaman, 1984). Yet, recent data may suggest that as many as 90 snails a day may be caught and eaten by radio-tagged juveniles (W. Kitchens, USGS, personal communication, July 12, 2005). However, this value appears high for two reasons: first, even if captures were done in only a few minutes (< 10 min; kites sometimes require 30-60 minutes; Bennetts and Kitchens, 2000), capture and extraction of 90 snails would require 15 hours; and second, based on the average snail weight reported by Eisemann et al. (1997), 90 snails would represent 774 grams of food or 174 percent of the kite's bodyweight. It is possible that the capture rate of 90 snails per day may have occurred in an area of unusually high snail density and that the snails were smaller than those collected by Eisemann et al. (1997). Because a conservative approach was deemed appropriate for this assessment, ingestion was modeled as triangular distribution with a range from 150.5 to 774 grams and 602 grams of snail per day as the most likely value, (i.e., approximately 70 snails).

A scoping-level assessment found dietary ingestion to be the most important pathway for MeHg uptake in the Everglades (i.e., as compared to water or incidental ingestion of sediment; Rumbold, 2004). In the present assessment, prey concentrations were derived from the District's permit-mandated biomonitoring program. This program required semiannual sampling of mosquitofish, and annual sampling of sunfish and largemouth bass at various sites within the STA and downstream Everglades (**Table 4**; for additional details on sampling design or analytical methods, see Rumbold, 2005a; 2005b).

Table 4. Summary of data sets (date of collection, number of fish) used to develop PDFs of mercury concentrations in fish tissues.

Location	Bass	Warmouth	Other Lepomids	Mosquitofish[a]
STA-2				
Supply Canal	10/04, n = 20	10/04, n = 1	10/04, n = 19	3/04, n = 1 10/04, n = 1
Cell 1[b]	10/04, n = 20	10/04, n = 11	10/04, n = 9	3/04, n = 2 10/04, n = 2
Cell 2	10/04, n = 20	10/04, n = 4	10/04, n = 16	3/04, n = 1 10/04, n = 1
Cell 3	10/04, n = 19	10/04, n = 0	10/04, n = 20	3/04, n = 1 10/04, n = 1
Discharge Canal	10/04, n = 20	10/04, n = 1	10/04, n = 19	3/04, n = 1 10/04, n = 1
WCA-1				
LOX F4	10/04, n = 20	10/04, n = 0	10/04, n=20	10/04, n = 1
ENP				
L67F1	10/04, n = 20	10/04, n = 1	10/04, n=19	10/04, n = 1

[a] Composite of 100–250 mosquitofish.

[b] Mosquitofish were collected from site C1A and C1X.

Use of empirical distribution functions to describe concentration distributions is based on the assumption of random, representative data sets and does not allow extrapolation beyond the range of observed data. However, electroshocking tends to result in samples biased towards larger fish (Reynolds, 1996). Moreover, the primary objective of the permit-mandated fish collections was for temporal and spatial trend analysis and did not target the preferred prey size range of these receptors. Accordingly, for the present assessment, probability density functions (PDFs) were developed, based on empirical distribution functions, and evaluated using goodness-of-fit tests (see documentation for Crystal Ball® software). Due to the limitations of these tests (for discussion, see USEPA, 1999), PDF selection relied heavily on graphical analysis and, where appropriate, defaulted to common distributions (e.g., lognormal for tissue concentrations).

Because eagles were not limited in prey size, mercury concentration data sets were simply parsed based on trophic level of fish (i.e., largemouth bass and warmouth as trophic level (TL) 4 and other sunfish as TL 3; TL classification based on information contained in Lange et al., 1998; Loftus et al., 1998; see **Table 5**). Alternatively, the storks' prey-size limitation required the data on mercury concentration in fish be preprocessed prior to developing PDFs (**Table 6**). Where datasets were sufficiently large, this involved statistical assessment for linear or non-linear relationships between fish size and tissue concentration (SigmaStat, Jandel Corporation, San Rafael, CA). For all data sets, but especially small data sets, this also involved graphical analysis. Where the size-Hg tissue concentration relationship was significant, fishes > 250 mm in length were censored prior to developing the PDFs for storks' exposures.

Extrapolations were used where data were not available for a given prey species (i.e., not collected as part of routine biomonitoring program). Tissue Hg concentrations for the Florida gar (*Lepisosteus platyrhincus*) were estimated based on an empirically derived proportional relationship with levels in concurrently collected largemouth bass reported by Loftus et al., 1998 (10 Florida gar and 24 largemouth bass from ENP) and Ted Lange (Florida Fish Wildlife Conservation Commission, personal communication, a total of 16 Florida gar and 34 largemouth bass from two WCA sites in 1997–1998). Likewise, tissue Hg concentrations in yellow bullheads (*Ictalurus natalis*) were based on a proportional relationship derived from 40 yellow bullhead and 24 largemouth bass collected from ENP (Loftus et al., 1998) and 5 yellow bullhead and 8 largemouth bass from WCA-3 in 1997 (T. Lange, personal communication). Although often feeding at TL 3 (Loftus et al., 1998), mosquitofish were used as TL 1 and TL 2 fish. To estimate Hg levels in apple snails at STA-2, an empirical, proportional relationship was derived from reports of Hg measurements in apple snails and concurrently collected mosquitofish. Loftus et al. (1998) determined Hg levels in 13 individual snails and 9 individual mosquitofish from Shark Slough. Rumbold (2000a) collected composite samples of snails (each comprised of 5–10 snails) and mosquitofish (each comprised of over 100 fish) at five different sites near Palm Beach County's North County Resource Recovery Facility (including the City of West Palm Beach Water Catchment Area). On average, snails contained 14 percent of the level of mercury that mosquitofish from the same area contained (ranged from 8 to 25 percent; modeled as a triangular distribution).

Table 5. Distributions of mercury levels (mg/kg) in prey species used to estimate exposure to bald eagles at various locations (unless otherwise noted, PDFs modeled using a lognormal distribution); for discussion of PDF selection, see page App. 4-6-12).

Location	Trophic Level 4 fish		Trophic Level 3 fish		Piscivorous birds
	2003	2004	2003	2004	
STA-2					
Supply Canal	0.18 ± 0.16	Logistic: mean = 0.09; scale = 0.04	0.09 ± 0.09	0.03 ± 0.02	TL3 x BAF BAF = 10.9 – 13.8, uniform
Cell 1	Weibull: loc. = 0.03, scale = 0.27, shape = 1.3	0.27 ± 0.14	Beta: A = 1.84, B = 9.33, scale = 1.2	Uniform: Min = 0.011, Max = 0.33	TL3 x BAF BAF = 10.9 – 13.8, uniform
Cell 2	Weibull: loc. = 0.05, scale = 0.13, shape = 1.2	0.14 ± 0.13	0.05 ± 0.02	0.05 ± 0.05	TL3 x BAF BAF = 10.9 – 13.8, uniform
Cell 3	0.12 ± 0.09	Logistic: mean = 0.06; scale = 0.014	0.03 ± 0.014	Tri: min = 0.004; likeliest = 0.016; max = 0.032	TL3 x BAF BAF = 10.9 – 13.8, uniform
Discharge Canal	0.55 ± 0.35	0.19 ± 0.17	Weibull: loc. = 0.01, scale = 0.13, shape = 0.9	0.07 ± 0.12	TL3 x BAF BAF = 10.9 – 13.8, uniform
WCA-1					
LOX F4	NA	0.21 ± 0.04	NA	0.1 ± 0.02	TL3 x BAF BAF = 10.9 – 13.8 uniform
ENP					
L67F1	0.76 ± 0.26	0.81 ± 0.42	0.36 ± 0.17	0.42 ± 0.31	TL3 x BAF BAF = 10.9 – 13.8 uniform

Table 6. Distributions of mercury levels (mg/kg) in prey species used to estimate exposure to wood storks at various locations (unless otherwise noted, PDFs modeled as lognormal; for discussion of PDF selection, see page App. 4-6-12).

Location	Florida Gar	Warmouth	Other Lepomids	Bass (Standard)	Yellow Bullhead	Small TL 1 & 2 fish[a]
STA-2						
Supply Canal	Standard bass x 0.43, 0.74, 0.96 (tri)[b]	Standard bass x 0.71 (point)[c]	0.03 ± 0.02	0.09 ± 0.07 (0.096 ± 0.04)	Standard bass x 0.5 – 0.78 (uniform)[d]	Uniform: 0.003 to 0.01
Cell 1	Standard bass x 0.43, 0.74, 0.96 (tri)[b]	0.23 ± 0.09	0.14 ± 0.11	0.24 ± 0.09 (0.32 ± 0.06)	Standard bass x 0.5 – 0.78 (uniform)[d]	Uniform: 0.01 to 0.029
Cell 2	Standard bass x 0.43, 0.74, 0.96 (tri)[b]	0.05 ± 0.01	0.05 ± 0.05	0.13 ± 0.2 (0.11 ± 0.07)	Standard bass x 0.5 – 0.78 (uniform)[d]	Uniform: 0.009 to 0.012
Cell 3	Standard bass x 0.43, 0.74, 0.96 (tri)[b]	Standard bass x 0.71 (point)[c]	Logistic: mean = 0.016, scale = 0.005	0.06 ± 0.04 (0.055 ± 0.02)	Standard bass x 0.5 – 0.78 (uniform)[d]	Uniform: 0.004 to 0.01
Discharge Canal	Standard bass x 0.43, 0.74, 0.96 (tri)[b]	Standard bass x 0.71 (point)[c]	0.07 ± 0.12	0.08 ± 0.02 (0.14 ± 0.14)	Standard bass x 0.5 – 0.78 (uniform)[d]	Uniform: 0.017 to 0.0173
WCA-1						
LOX F4	Standard bass x 0.43, 0.74, 0.96 (tri)[b]	Standard bass x 0.71 (point)[c]	0.1 ± 0.02	0.19 ± 0.06 (0.22 ± 0.04)	Standard bass x 0.5 – 0.78 (uniform)[d]	Point: 0.048
ENP						
L67F1	Standard bass x 0.43, 0.74, 0.96 (tri)[b]	Standard bass x 0.71 (point)[c]	0.42 ± 0.3	0.48 ± 0.18 (0.55 ± 0.09)	Standard bass x 0.5 – 0.78 (uniform)[d]	Point: 0.05

[a] Based on Hg levels in mosquitofish.

[b] Based on relationship derived from levels reported by Loftus et al., 1998 (10 Florida gar and 24 largemouth bass from the ENP); T. Lange (personal communication, a total of 16 Florida gar and 34 largemouth bass from two sites within WCA-3A in 1997–1998).

[c] Based on the relationship between warmouth and largemouth bass collected from STA-2, Cell 1 in 2004.

Standard bass is 250 mm whole-body concentration.

[d] Based on relationship derived from levels reported by Loftus et al., 1998 (40 yellow bullhead and 24 largemouth bass from the ENP); T. Lange (personal communication, 5 yellow bullhead and 8 largemouth bass from WCA-3 in 1997).

Tissue Hg concentrations varied greatly in fish from different locations within the STA. Therefore, exposure from each component was modeled separately and recombined using area use factors (AUf). As discussed in Rumbold (2004), ideally these AUfs would have been based on ecological models that considered the various combinations of environmental characteristics that determine an area's suitability for a given species. Among others, these characteristics include water depth, vegetation density, and densities and size distribution of the preferred prey populations (Surdick, 1998; Gawlik, 2002). Water depth and vegetation density are key factors in defining the physical environment and, thus, habitat suitability for the predator, but also for the prey population. For example, optimal forage habitat for wood storks is relatively calm water, where depths are between 10 and 25 cm, and where the water is uncluttered by dense patches of aquatic vegetation (Coulter and Bryan, 1993, see http://www.fws.gov/verobeach/species/birds/wost/wost-msrp/wost_behavior.htm). Suitable foraging habitat for the snail kite is typically a combination of low profile (< 3 m) marsh with an interdigitated matrix of shallow (0.2–1.3 m deep) open water, which is relatively clear and calm. Snail kites require foraging areas that are relatively clear and open in order to visually search for apple snails. Therefore, dense growth of herbaceous or woody vegetation is not conducive to efficient foraging (USFWS, 1999). Eagles would also require relatively open water when foraging for fish. Accordingly, Cell 1, much of which is densely vegetated (**Figure 2**), may not be optimal habitat for these three species.

Developing habitat suitability models that would capture all possible combinations of these factors was beyond the scope of this assessment (also, see discussion in the *Model Validation* section). Instead AUfs were assigned based on surface area at average stage, as follows: supply canal (27.4 ha) - 0.01, Cell 1 (805 ha) - 0.305, Cells 2 and 3 (both at 898 ha) each - 0.341, and the discharge canal (8.5 ha) - 0.003. To simplify interpretation, AUfs were held constant. However, because fish from Cell 1 and the discharge canal generally had the highest mercury levels in 2003, a worst-case scenario, i.e., of a bird foraging exclusively in Cell 1 or the small discharge canal, was simulated separately using an AUf of 1.

EFFECTS ANALYSIS

The first step in characterizing effects is to describe the effects elicited by the stressor and then evaluate them in terms of a potential link with the assessment endpoint.

The link between the assessment endpoint in the present analysis (i.e., reproduction and recruitment of the birds) and mercury has been reviewed elsewhere (Barr, 1986; Heinz, 1996; Thompson, 1996; Wolfe et al., 1998; see also Heinz and Hoffman, 2003; Wiener et al., 2003). The present effects analysis is based on the work of Heinz (1979) in which he dosed three generations of mallards (*Anas platyrhynchos*) at 0.5 mg/kg food and observed statistically significant effects in both the pre-nesting female and the hatchlings she produced. In terms of mean measurements from the three mallard generations combined, Heinz (1979) reported that the percent of eggs laid outside the nest box increased from 4.3 percent in the controls to 9.7 percent in the dosed group, percent of ducklings approaching calls decreased from 97 percent in the controls to 94 percent in the dosed group, distance traveled by chicks in avoidance increased from 32 cm in the controls to 38 cm in the dosed group and, finally, the number of 1-week-old nestlings produced decreased from 46 in the controls to 37.5 in the dosed group.

Results from Heinz (1979) have been used previously in deriving an acceptable daily MeHg dose (USEPA, 1995, 1997; Windward Environmental LLC, 2003; USFWS, 2003; Rumbold, 2004; 2005a). For the Great Lakes Initiative, the U.S. Environmental Protection Agency

(USEPA, 1995) estimated 0.064 mg MeHg/kg bw/day as the lowest-observed-adverse-effect level (LOAEL) from the work of Heinz (1979). USEPA later adjusted this dose rate upward to 0.078 mg MeHg/kg bw/day based on the food consumption by dosed hens rather than control hens, which consumed less (USEPA, 1997). Dividing this LOAEL by an uncertainty factor (UFL) of 3, USEPA (1997) derived a chronic no-observed-adverse-effect level (NOAEL, 0.026 mg MeHg/kg bw/day) for use as the toxicity reference value (TRV) to protect the pre-nesting female and embryo.

RISK CHARACTERIZATION

Risk characterization is the process of estimating the potential risks to the receptors by integrating exposure and effects data under a specific set of conditions.

BALD EAGLE

Rumbold (2004; 2005a) reported simulations of daily intake of MeHg for an eagle could range up to 0.082 mg Hg/kg BW/day, if foraging over the entire STA, or as high as 0.75 mg Hg/kg BW/day, if foraging exclusively from the discharge canal. Monte Carlo simulations in the present assessment resulted in substantially lower exposures (**Figure 5**).

The reduction in simulated exposure was a direct result of lower Hg levels in fish collected in 2004 as compared to previous years (**Table 5**; **Figures 6** and **7**). This decline was most evident in bass from the discharge canal (Figures 6) that exhibited significant among-year differences in Hg levels (ANCOVA, df = 3,75; $F = 29.06$, $p < 0.001$); though pair-wise comparisons found no difference in LSM between 2001 and 2002 (Tukey-Kramer post-hoc test, $p = 0.83$), levels in both 2004 and 2003 differed from levels in 2001 and 2002 ($p < 0.0001$), and 2004 differed from 2003 ($p < 0.0001$). This relatively rapid change in Hg levels in large-bodied fish, likely owing to changes in population structure, is not surprising given the dynamic nature (harsh environment) of this small canal during extreme pump operations (stage in canal can be drawn down 6 feet). Hg levels have also declined in bass collected from site C1X in 2004 (**Figure 6**), though the decline appears much more gradual than in the canal population. This is most evident in the young bass (**Figure 6**) suggesting recent declines; levels of Hg in 1-year-old bass varied among years (ANOVA, df = 2, 22; $F = 4.2$, $p = 0.03$) with mean concentrations of 0.56 mg/kg in 2001, no fish in 2002, 0.49 mg/kg in 2003, and 0.34 mg/kg in 2004 (Tukey Post-hoc test found 2004 fish differed significantly from 2001 but not 2003 fish).

Results from monitoring sunfish were mixed in terms of temporal tends. In part, this may be a function of the difficulty in unraveling two confounding factors: (1) size of sunfish varied (i.e., fish size is a surrogate for age and Hg levels in fishes tend to increase with age), and (2) multiple lepomid species were collected and Hg levels have been found to vary among species (as opposed to just age in the bass). When warmouth were singled out and normalized to total length, Hg levels at site C1X differed among years (df = 2, 20; $F = 27.9$, $p < 0.001$) with 5.45 mg/kg/cm in 2002 fish, 2.57 mg/kg/cm in 2003 fish, and 1.72 mg/kg/cm in 2004 fish (Tukey Post-hoc test found both 2004 fish and 2003 fish differed significantly from 2002 but not between 2004 and 2003). When a similar analysis was carried out on red ear sunfish from site C1X, fish from both 2004 and 2003 again contained significantly lower Hg levels than fish collected in 2002 (Kruskal-Wallis test; $H = 15.1$, df = 3, $p = 0.002$; Dunn's test $p < 0.05$). Similar to the warmouth, 2004 fish did not differ from 2003 fish.

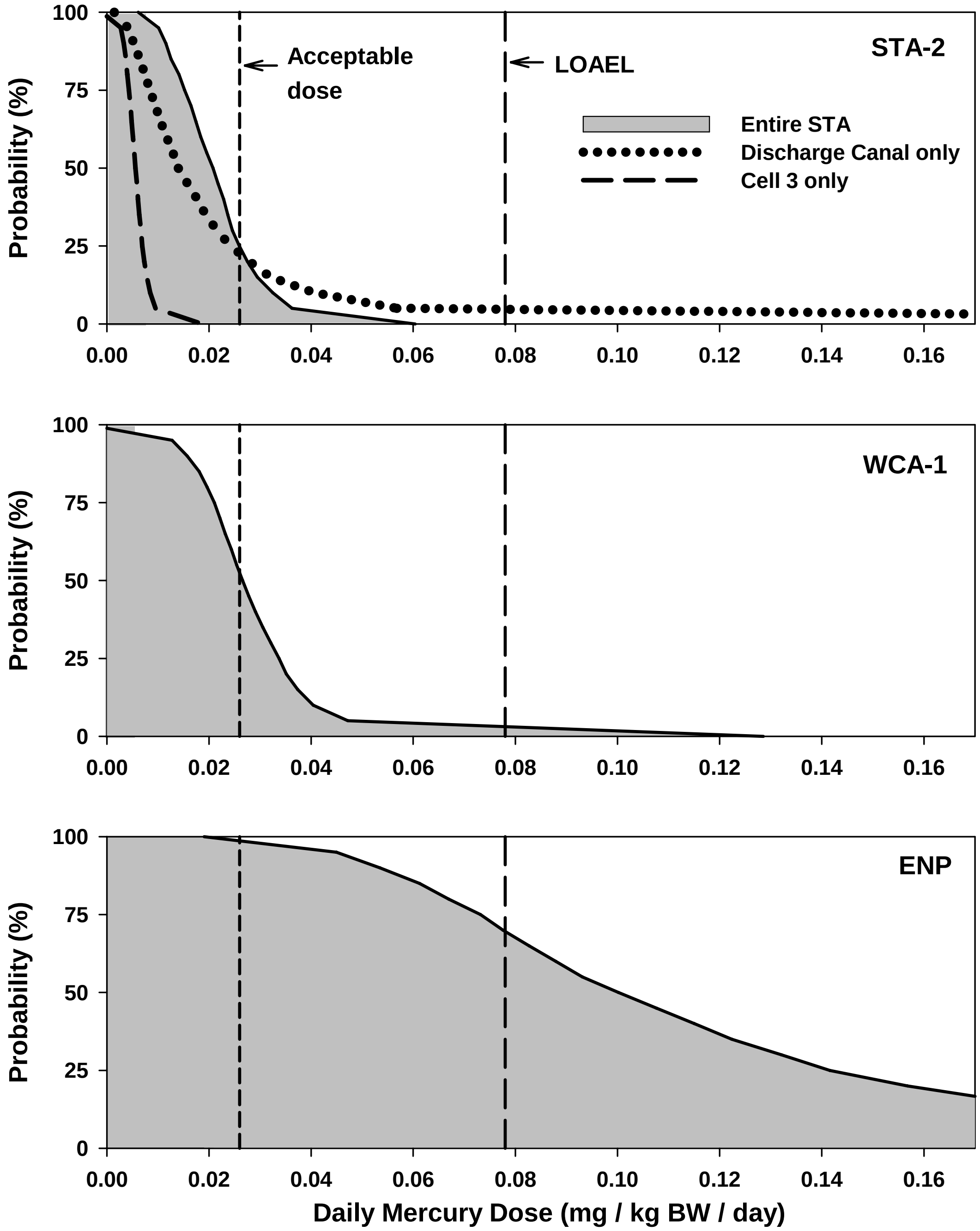


Figure 5. Comparison of reverse cumulative probability distributions of simulated daily MeHg intake by pre-nesting female eagles foraging at various locations to the mallard LOAEL and NOAEL-based TRV (i.e., acceptable dose). The ordinate value (y) represents the probability of exceeding the daily mercury intake on the abscissa (x).

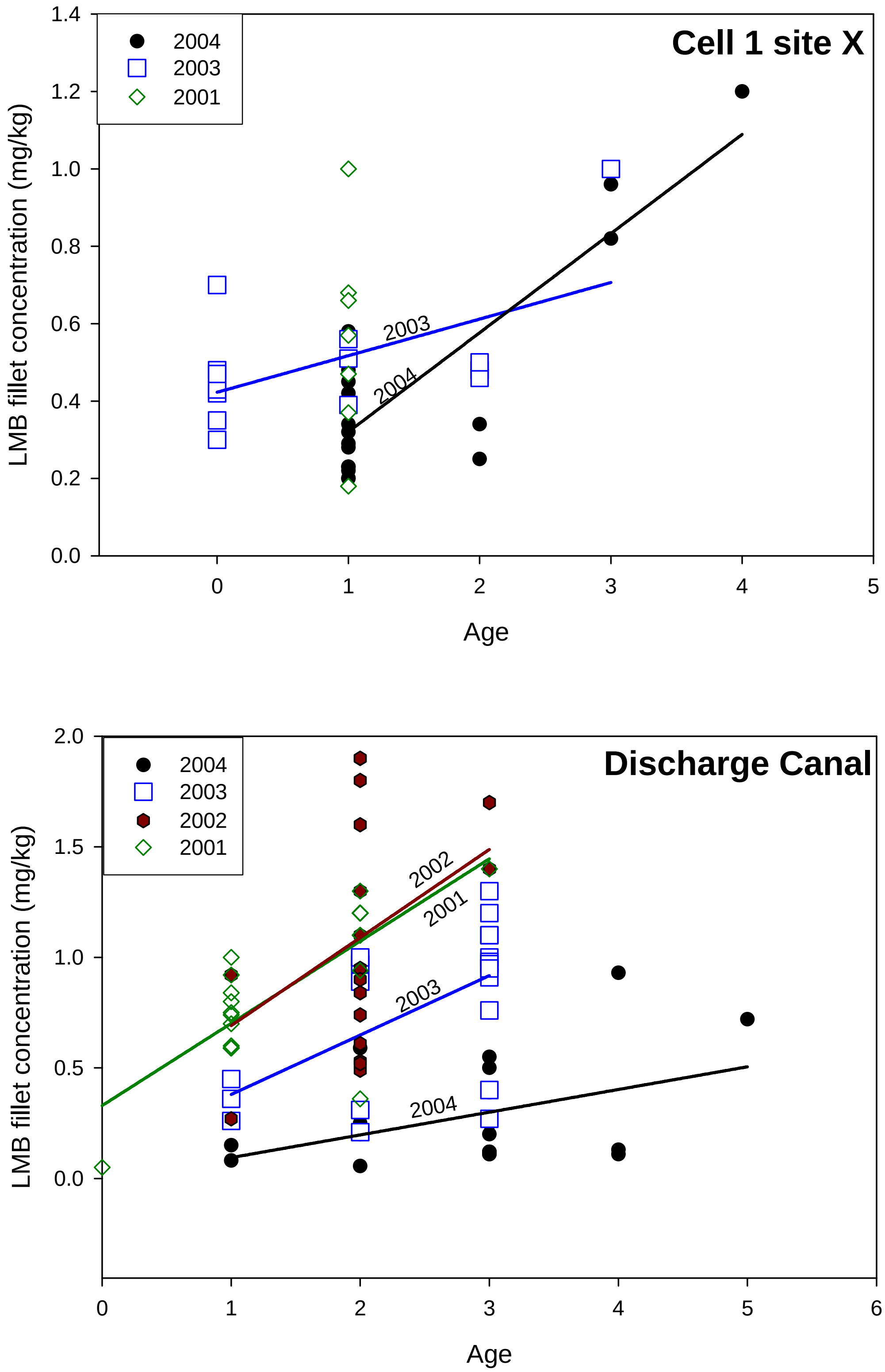
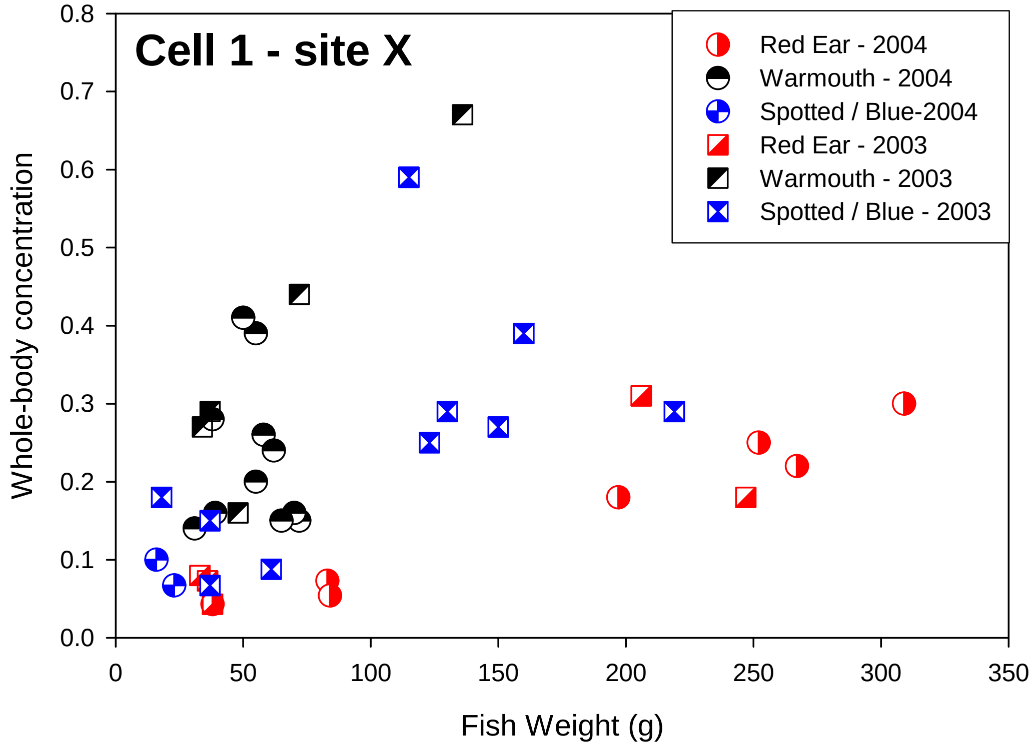


Figure 6. Change in Hg levels in largemouth bass (normalized to age) from STA-2, Cell 1 and the discharge canal over time.



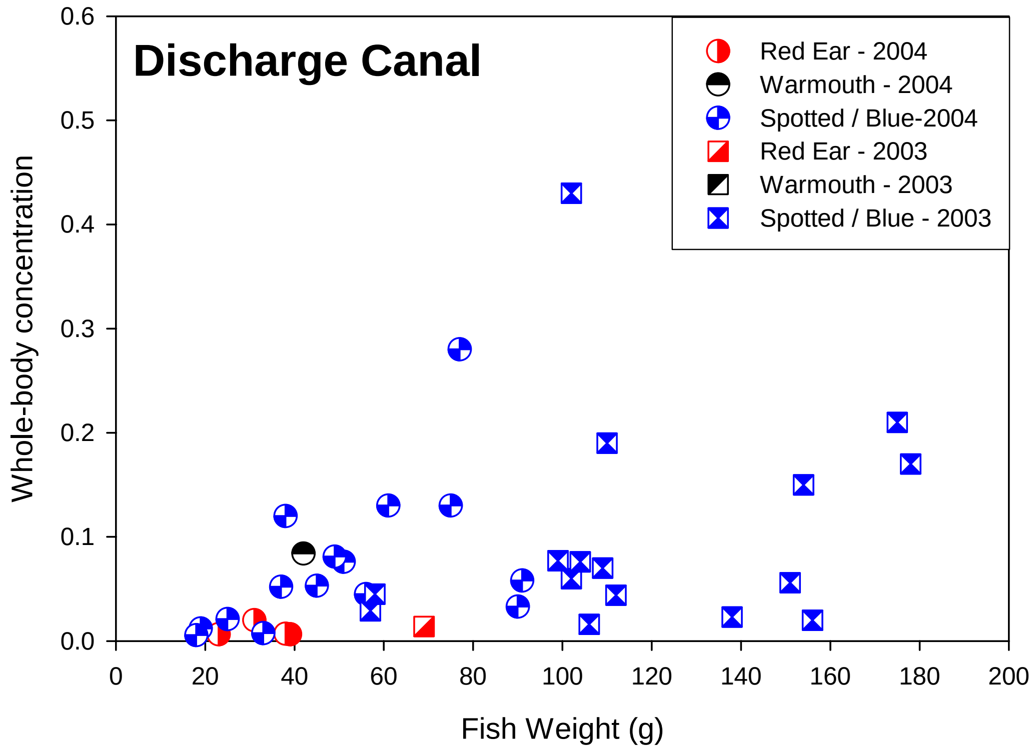


Figure 7. Hg levels in sunfish (normalized to weight) from STA-2, Cell 1 and the discharge canal.

Interestingly, unlike the bass in the Discharge Canal that showed a clear decline, resident bluegill from the discharge canal (most numerous species collected over the monitoring period) did not exhibit among-year differences in Hg levels (for the period from 2001–2004; ANOVA, df = 3, 32; F = 1.41; p = 0.26).

Although the decline in Hg levels in bass from the canal dramatically reduced simulated worst-case exposure to an eagle (i.e., scenario where they were foraging only from the canal), owing to its relative small area relative to the treatment cells, the decline here did not significantly lessen the exposure to a bird, when integrated over the entire site. On the surface, it appears that the gradual downward trend in mercury in fish from site C1X was not sufficient to lower the exposure reported by Rumbold (2004; 2005a). However, there was also a between-year difference in sampling that may confound this interpretation. Fish were collected from two different locations in Cell 1 in 2001 and 2002 (for details, see Rumbold, 2004). To evaluate temporal trends both locations were then sampled in 2003 (n = 34 bass and n = 36 sunfish). Rumbold (2004) utilized the entire the data set as being representative of exposure from the entire cell because Cell 1 was known to exhibit a strong north-south gradient in mercury, and one of two sites was in the north while the other was in the south. In 2004, only the southern site, C1X, was sampled and consequently the current assessment had data available only from the area that, at one point, was the "MeHg hotspot."

The likelihood that these simulated exposures would be harmful to the eagle was assessed by superimposing the LOAEL and its derived NOAEL-based TRV (Heinz, 1979; **Figure 5**). The results indicate that the probability of an eagle experiencing exposures above the TRV was 28 percent, if foraging over the entire STA. The probability of that same eagle exceeding the LOAEL was negligible.

WOOD STORK

Probability distributions of simulated daily MeHg intake by wood storks are shown in **Figure 8**. When the distributions were compared to the effects thresholds, a stork foraging over the entire STA would have less than 1 percent probability of experiencing exposures above the TRV. On the other hand, storks foraging exclusively in Cell 1 would have a 54 percent probability of experiencing a daily intake exceeding the TRV. As discussed previously, owing to the poor foraging habitat in Cell 1, this may not be an accurate representation of true risk. Simulated exposures never exceeded the LOAEL under any of the foraging scenarios within STA-2.

As was seen for the eagle, simulations of daily MeHg intake for the wood stork were very sensitive to differences in Hg levels in fish. This resulted in highly variable risk, depending on where the bird foraged within the Everglades (**Figure 8**).

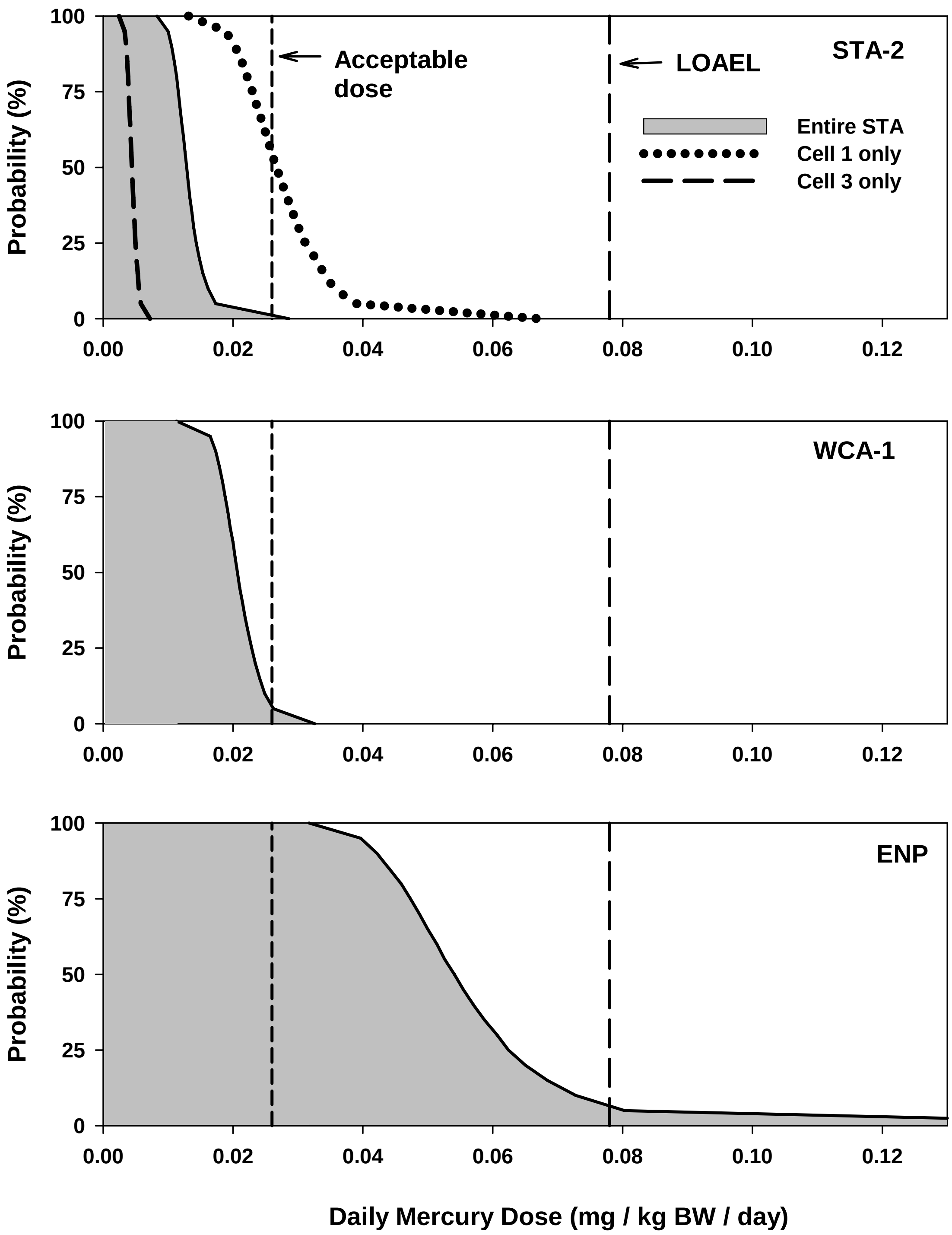


Figure 8. Reverse cumulative probability distributions of simulated daily MeHg intake by pre-nesting adult wood storks.

SNAIL KITE

Simulated daily MeHg intake by snail kites feeding on apples snails at STA-2 resulted in only negligible risk, far below the NOAEL-based TRV (**Figure 9**).

Apple snails collected from South Florida have been reported to contain mercury levels as high as 0.35 mg/kg (wet weight; Rumbold, 2000a). However, those snails were collected from a stormwater detention area that had recently been contaminated with fly ash from a municipal solid-waste combustor and, as such, did not represent typical ambient conditions. Average concentration in snails in that study, which included four other sites, was 0.035 ± 0.4 mg/kg (Rumbold, 2000a) and was comparable to averages reported by Eisemann et al. (1997; mean of 0.063 mg/kg in 62 snails collected from Central and South Florida) and Loftus et al. (1998; 0.019 ± 0.008 mg/kg in 13 snails collected from Shark River Slough). It should be noted that at the time those snails were collected, mercury levels were thought to be peaking in other taxa (e.g., fish, birds, alligators) in the Everglades. Although these levels appear low, they are considerably higher than Hg levels reported in *Pomacea sp.* collected from the Pantanal region of South America (0.0005 mg/kg, Leady and Gottgens, 2001).

Based on the proportionality with concentrations measured in mosquitofish (highest concentration in mosquitofish was 0.029 mg/kg; **Table 4**), the maximum forecasted Hg concentration in apple snails in the present assessment was 0.0073 mg/kg. It should be noted that, in 2004, mosquitofish from STA-2 contained relative little Hg relative to mosquitofish from other areas of the Everglades (50^{th} and 95^{th} percentile concentrations are 0.07 and 0.241 mg/kg, respectively, in mosquitofish collected by District at Everglades marsh sites over the period of record; Rumbold, 2005b). Therefore, kites feeding on apple snails were at relatively lower risk of mercury biomagnification and intoxication than the piscivorous birds. This finding that mercury biomagnification along herbivorous links poses lower risk is consistent with results from other studies (Eisemann et al., 1997; Leady and Gottgens, 2001).

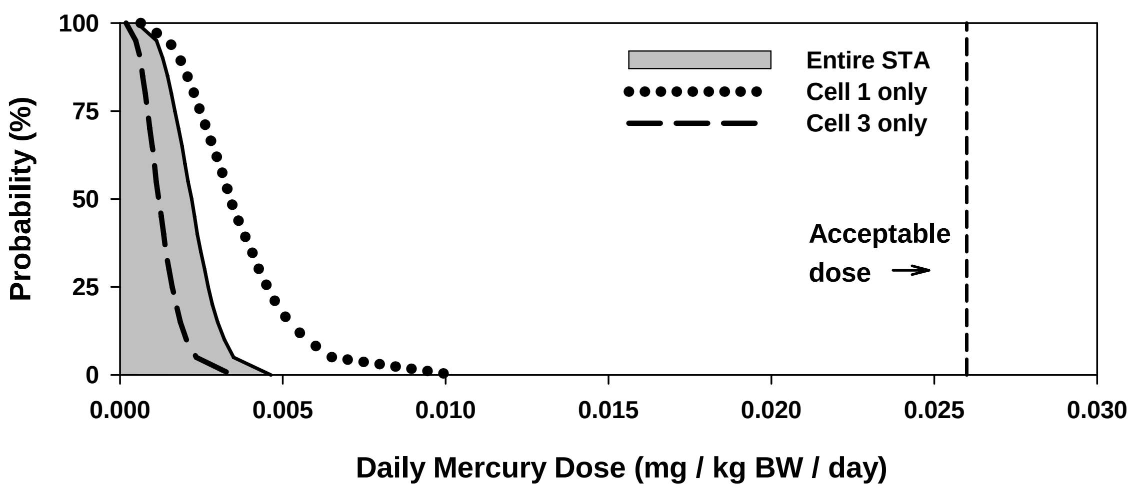


Figure 9. Reverse cumulative probability distributions of simulated daily MeHg intake by pre-nesting adult snail kites foraging at STA-2.

MODEL VALIDATION

Because much more information was available on great egrets as compared to the other receptors, previous assessments (Rumbold 2000b; 2004; 2005a) utilized results from the egrets to validate exposure models and resulting risk estimates, then extrapolated those findings to the other receptors. First, egret eggs and feathers were being collected at two colonies in WCA-3 to monitor mercury trends. Comparing these measured concentrations to predicted concentrations from simulations in WCA-3 enabled the validation of the exposure model. When this comparison was made, model-predicted egg and nestling feather mercury levels were found to exceed measured concentrations (by a factor of 1.4, Rumbold, 2000b; by 24 percent and 6 percent, Rumbold, 2004), thus suggesting the dietary-consumption model overestimated exposure to the birds. This overestimation was thought to result from conservatism in several of the embedded assumptions or methods, in particular: (1) the bias of electrofishing towards collection of larger fish that skewed mercury distributions to the right (i.e., high), and (2) the assumption of 100 percent contact time. The latter is a fundamental limitation in the simplified approach to estimate exposure in these wide-ranging receptors that will always overestimate the effect-size for any given action. Rumbold (2004; 2005a) assumed 100 percent contact time for bald eagles (i.e., foraged only at STA-2). This was appropriate at that time, owing to the fact that the eagles were nesting at the STA and because eaglets remain and forage near the natal nest until 21 weeks of age (Wood et al., 1998); however, because the eagles no longer nested at the STA, this assumption was not an accurate representation in the present assessment. Likewise, the assumption in the present assessment of 100 percent contact time for the wood stork, which is known to be wide-ranging, was not accurate representation. Based on the maps shown in **Figures 3** and **4**, the proportion of potentially affected area (i.e., STA-2, Cell 1) was small relative to the area of the range of these receptors. To better estimate exposure would have required the development of an individual-based movement (IBM) model with “rules of movement” describing how an individual receptor would respond to variations in the landscape (for an example, see Hope, 2005). However, this was beyond the scope (and resources) of the current assessment.

Measured concentrations in the egret eggs and feathers from the colonies in WCA-3 also allowed for a comparison of the risk estimate based on daily consumption to risk estimates based on literature-derived, critical tissue concentrations reported by Thompson (1996) and Spalding et al. (2000). As expected, given the discussion above, simulated daily MeHg intake and Heinz’s reference doses were found to overstate the risk to the birds when compared to risks based on critical tissue concentrations (Rumbold 2000b; 2004; 2005a).

Finally, risk estimates derived from simulations were also evaluated based on observed reproductive success of egrets in WCA-3 (Rumbold 2000b; 2004; 2005a). Except for the past two years, in which unseasonably heavy rainfall and sudden increases in water levels have resulted in poor nesting, egrets have enjoyed high reproductive success in the Everglades (Frederick et al., 1997; 1999; Gawlik, 1999; 2002). Therefore, it was argued that the daily consumption model overstated the risk to birds in WCA-3, at least in terms of population-level effects, and thus, a lower risk category should be acceptable elsewhere (Rumbold, 2000b). Overstated risk was thought to be a result of: (1) exposure overestimation and (2) the small effect-size observed by Heinz (1979; for discussion, see Rumbold 2004; 2005a).

Regrettably, data on mercury levels in bald eagles, wood storks and snail kites in South Florida is extremely limited and what little information is available is outdated, thus model validation could not be carried out for these receptors. There is also no way to know for certain that exposure and risk would be overestimated and overstated as was found in previous assessments for egrets using similar models. However, it should be noted that, as with the great egret, fluctuations in populations of these receptors do not correlate with temporal trends observed in Hg levels. Great egrets began to show their progressive increase in breeding numbers in the 1991–1993 period (for review, see Crozier and Cook, 2004), at a time when Hg levels were thought to be high. At this same time, Florida supported the highest number of breeding bald eagles of any southeastern state, supporting approximately 70 percent of the occupied territories in this region (Nesbitt, 1995, as cited by http://www.fws.gov/verobeach/species/birds/baea/baea-msrp/baea-status_trends.htm). Furthermore, during this same time period, the kite population was also at a high, but has subsequently declined (in 2003 the population size was estimated to be half that of 1999; Brandt, 2005), at a time when Hg levels were on the decline (for discussion of Hg declines, see Atkeson and Axelrod, 2004). However, this represents only a crude assessment of population-level effects. It does not rule out individual-level effects, particularly for receptors foraging in the remaining MeHg hotspots (e.g., ENP, see bottom panel in **Figures 5** and **9**).

RISK MANAGEMENT GOALS

In human health assessments, the risk management goal is to protect the individual, even the high end or what is known as the reasonable maximal exposure (RME) individual – often taken to be the 90th or 95th percentile. By comparison, in most ecological assessments, the risk management goal is to guard against population-level effects by protecting some hypothetical "average" individual or what is called the central tendency exposure individual (i.e., the 50th percentile). However, in cases such as this, where the ecological receptor is a threatened or endangered species it may be more appropriate to protect the RME individual (for differences in designated protection, review ESA section 4 versus section 9). Thus, the goal would be to have simulated exposure for the RME individual positioned to the left of (i.e., below) the NOAEL-based TRV. Nonetheless, there remains no established default decision threshold for identifying what level of risk is either clearly acceptable or clearly unacceptable for a RME (Suter, 1993; USEPA, 1999); nor should there be for several reasons.

First, use of "bright lines" (i.e., single value thresholds or decision criteria above which the risk is considered unacceptable by definition) presumes a level of accuracy in both exposure models and effects measures that does not exist in practice (NRC, 1994; Presidential/Congressional Commission on Risk Assessment and Risk Management, 1997). Risk estimates must be interpreted relative to the assumptions on which the assessment was based (see preceding discussion of model validation). Obviously, the more conservative the exposure and effects assumptions, the broader the range of tolerable risk.

Under certain circumstances, it may also be necessary to differentiate and accept some level of background risk and assess only the excess risk due to some new action. Applying a ridged risk management goal for mercury exposure to Everglades' wildlife is made much more difficult because it is a regional problem driven, at least in part, by a factor outside of the control of local managers (i.e., atmospheric deposition). Therefore, some degree of risk from mercury exposure may be uncontrollable and, thus, unavoidable. The difficulty is defining exactly what the background risk is. In the present assessment, simulations were done for birds foraging in Loxahatchee National Wildlife Refuge (site LOXF1 in WCA-1; see middle panel in **Figures 5** and **9**) as a possible indication of background risk. The next step would be for risk managers to decide what site- or action-contributed increment of ecological risk, relative to this background risk, is acceptable.

Finally, risk managers may sometimes elect to tolerate greater risk, if there is a favorable risk-benefit ratio, a poor cost-benefit ratio or both (e.g., see Federal Insecticide, Fungicide, and Rodenticide Act; Comprehensive Environmental Response, Compensation, and Liability Act;

also see beneficial effects in USFWS, 1998). However, the flexibility afforded to risk managers to tolerate greater risk varies by agency and also by legislative mandate (Schierow, 1998). Further, there is often little guidance provided for net environmental benefits analysis (for review, see Efroymson et al., 2003) resulting in a wide range of different interpretations and applications.

EXTRAPOLATIONS TO FUTURE CELL 4

The objective of the present assessment was to provide supplemental information for a permit application for the expansion of STA-2 into Cell 4. To do this required: (1) updating the current status of mercury risk at STA-2 and (2) extrapolating this current risk to wildlife foraging in Cell 4 in the future.

It is beyond the scope of this risk assessment to discuss in detail all the complexities in the biochemical processes in sediment and water that control net mercury methylation in the Everglades (for reviews, see Gilmour and Krabbenhoft, 2001; Gilmour et al., 2004) or that account for the observed among-cell differences in STA-2 (Fink, 2005; Rumbold and Fink, in press). Although the precise biogeochemistry has yet to be confirmed, the current theory holds that Cell 1 sediments had redox conditions and sulfate concentrations that stimulated microbial methylation, whereas Cell 3 did not. Further, the conditions that made Cell 3 less favorable to mercury methylation, i.e., sulfide buildup and methylation inhibition (for discussion of sulfide inhibition, see Benoit et al., 1999a; 1999b) were thought to be a result of the prior land use, i.e., farming. This may also explain why STA-1W, which was farmed and fertilized prior to being reclaimed as a constructed wetland, did not have a mercury problem (Fink, 2003). Because they share similar prior land use and similar operational designs (current design of Cell 4 calls for submerged aquatic vegetation similar to Cell 3; Brown and Caldwell, 2005), Cell 3 is the most appropriate reference for extrapolation to future Cell 4 conditions.

To this end, simulations of MeHg exposure and risk were carried out for receptors, if foraging only in Cell 3 (**Figures 5**, **8**, and **9**). The results demonstrate that the RME individual is well below the NOAEL-based TRV (i.e., acceptable dose) for all three receptors. Nonetheless, a risk management option that is likely to be required to offset any remaining uncertainty in the assessment is to monitor mercury levels in Cell 4 and to have in place adaptive managements strategies with decision criteria (i.e., if-then statements) to respond to undesired responses.

CONCLUSIONS

Risk characterization indicated that the likelihood that MeHg exposures to bald eagles, wood storks, or snail kites foraging currently at STA-2 would exceed effects thresholds was low at the time of this assessment and, in the case of the bald eagle, has decreased since 2003. This places these birds in a low-risk category, given the assumptions on which the assessments were based. Even without consideration of background risk or the risk-benefit ratio, this risk category appears acceptable. Further, if current conditions in Cell 3 are a valid reference for predicting future conditions in Cell 4, mercury risks to birds foraging in Cell 4 would appear to be negligible. Any remaining uncertainty in the assessment can be offset through a risk management option of monitoring mercury levels in Cell 4 and by having adaptive management strategies with explicit decision criteria in place to respond to undesired, unpredicted outcomes.

LITERATURE CITED

Atkeson, T. and D. Axelrod. 2004. Chapter 2B: Mercury Monitoring, Research and Environmental Assessment. G. Redfield, ed. In: *2004 Everglades Consolidated Report*, South Florida Water Management District, West Palm Beach, FL.

Barr, J.F. 1986. Population Dynamics of the Common Loon (*Gavia immer*) Associated with Mercury-Contaminated Waters in Northwestern Ontario. *Can. Wildl. Serv.* Occas. Pap. No. 56: 23.

Beissinger, S.R. 1983. Hunting Behavior, Prey Selection, and Energetics of Snail Kites in Guyana: Consumer Choice by a Specialist. *Auk,* 100: 84-92.

Beissinger, S.R. 1990. Alternative Foods of a Diet Specialist, the Snail Kite. *Auk*, 107: 327-333.

Bennetts, R.E. and W.M. Kitchens. 2000. Factors Influencing Movement Probabilities of a Nomadic Specialist: Proximate Foraging Benefits or Ultimate Gains from Exploration? *Oikos*, 91: 459-467.

Benoit, J.M., C.C. Gilmour, R.P. Mason and A. Heyes. 1999a. Sulfide Controls on Mercury Speciation and Bioavailability to Methylating Bacteria in Sediment Pore Waters. *Env. Sci. Technol.*, 33(6): 951-957.

Benoit, J.M., R.P. Mason and C.C. Gilmour. 1999b. Estimation of Mercury-Sulfide Speciation in Sediment Pore Waters Using Octanol-Water Partitioning and Its Implications for Availability to Methylating Bacteria. *J. Env. Toxicol. Chem.*, 8(10): 2138-2141.

Bortolotti, G.R. 1984. Physical Development of Nestling Bald Eagles with Emphasis on the Timing of Growth Events. *Wilson Bull.*, 96: 524-542.

Brandt, L.A. 2005. Summary of Arthur R. Marshall Loxahatchee National Wildlife Refuge 2004 Science Workshop. U.S. Fish and Wildlife Service, Arthur R. Marshall Loxahatchee Natural Wildlife Refuge, Boynton Beach, FL.

Brown and Caldwell. 2005. Stormwater Treatment Area 2 Cell 4 Expansion Project. Basis of Design Report. CN040935-WO04. Report to South Florida Water Management District, West Palm Beach, FL.

Bryan, A.L., Jr., and M.C. Coulter. 1987. Foraging Flight Characteristics of Wood Storks in East-Central Georgia, USA. *Colonial Waterbirds*, 10: 157-161.

Coulter, M.C. and A.L. Bryan, Jr. 1993. Foraging Ecology of Wood Storks (*Mycteria americana*) in East-Central Georgia. I. Characteristics of Foraging Sites. *Colonial Waterbirds*, 16: 59-70.

Crozier, G.E. and M.I. Cook (eds.). 2004. South Florida Wading Bird Report. Volume 10. Unpublished Report. South Florida Water Management District, West Palm Beach, FL. Online at www.sfwmd.gov/org/wrp/wrp_evg/projects/2_wrp_evg_projects.html#wading [November 2004].

Crozier, G.E. and D.E. Gawlik, (eds.) 2003. South Florida Wading Bird Report. Volume 9. Unpublished Report. South Florida Water Management District, West Palm Beach, FL. Online at www.sfwmd.gov/org/wrp/wrp_evg/projects/2_wrp_evg_projects.html#wading [March 26, 2004].

Darby, P.C., P.L. Valentine-Darby, R.E. Bennetts, J.D. Croop, F. Percival and W.M. Kitchens. 1997. Ecological Studies of Apple Snails (*Pomacea paludosa, Say*). Report to the South Florida Water Management District, West Palm Beach, FL.

Depkin, F.C., M.C. Coulter and A.L. Bryan Jr. 1992. Food of Nestling Wood Storks in East-Central Georgia. *Colonial Waterbirds*, 15: 219-225.

Efroymson, R.A., J.P. Nicolette and G. Suter II. 2003. A Framework for Net Environmental Benefit Analysis for Remediation or Restoration of Petroleum-Contaminated Sites. U.S. Department of Energy National Petroleum Technology Office.

Eisemann, J.D., W.N. Beyer, R.E. Bennetts and A. Morton. 1997. Mercury Residues in South Florida Apple Snails (*Pomacea paludosa*). *Bull. Environ. Contam. Toxicol.*, 58: 739-743.

Fink, L.E. 2002. Status Report on the Effect of Water Quantity and Quality on Methylmercury Production. Appendix 2B-2. In: 2002 Everglades Consolidated Report. South Florida Water Management District, West Palm Beach, FL. Online at http://www.sfwmd.gov/org/ema/everglades/index.html [March 26, 2004].

Fink, L.E. 2003. The Effect of Dryout and Burn on the Everglades Mercury Cycle. Appendix 2B-1. In: 2003 Everglades Consolidated Report, South Florida Water Management District, West Palm Beach, FL. Available online at http://www.sfwmd.gov/org/ema/everglades/ index.html [July 25, 2005].

Fink, L.E. 2004. STA-2 Mercury Special Studies Interim Report. Appendix 2B-7. In: 2004 Everglades Consolidated Report. South Florida Water Management District, West Palm Beach, FL. Available online at http://www.sfwmd.gov/org/ema/everglades/index.html [March 26, 2004].

Fink, L.E., J. King, P. Adak and F. Matson. 2005. Appendix 2B-2: STA-2 Mercury Special Studies Project Report. G. Redfield, ed. In: *2005 South Florida Environmental Report – Volume I*. South Florida Water Management District. West Palm Beach, FL.

Frederick, P.C. and M.W. Collopy. 1988. Reproductive Ecology of Wading Birds in Relation to Water Conditions in the Florida Everglades. Report to U.S. Army Corps of Engineers, Jacksonville, FL.

Frederick, P.C., Fontaine, P., Heath, J., Babbitt, G. 1999. Factors affecting breeding status of wading birds in the Everglades. Report to the U.S. Army Corps of Engineers. Research Work Order No. 188. Jacksonville, FL.

Frederick, P.C., M.G. Spalding, M.S. Sepulveda, G.E. Williams Jr., S. Bouton, H. Lynch, J. Arrecis, S. Loerzel and D. Hoffman. 1997. Effects of Environmental Mercury Exposure on Reproduction, Health and Survival of Wading Birds in Florida Everglades. Report to Florida Department of Environmental Protection, Gainesville, FL.

Gariboldi, J.C., C.H. Jagoe and A.L. Bryan Jr. 1998. Dietary Exposure to Mercury in Nestling Wood Storks (*Mycteria americana*) in Georgia. *Archives of Environmental Contamination and Toxicology*, 34: 398-405.

Garrett, M.G., J.W. Watson and R.G. Anthony. 1992. Bald Eagle Home Range and Habitat Use in the Columbia River Estuary. *Journal of Wildlife Management*, 57, 19-27. 1993.

Gawlik, D.E. (ed.) 1999. South Florida Wading Bird Report. Vol. 5, Issue 1. South Florida Water Management District, West Palm Beach, FL. Available online at http://www.sfwmd.gov/org/wrp/wrp_evg/projects/2_wrp_evg_projects.html#wading [March 26, 2004].

Gawlik, D.E. (ed.) 2002. South Florida Wading Bird Report. Vol. 8, Issue 1. South Florida Water Management District, West Palm Beach, FL. Available online at http://www.sfwmd.gov/org/wrp/wrp_evg/projects/2_wrp_evg_projects.html#wading [March 26, 2004].

Gilmour, C.C., and D.P. Krabbenhoft. 2001. Status of Methylmercury Production Studies. Appendix 7-4. In: 2001 Everglades Consolidated Report, South Florida Water Management District, West Palm Beach, FL. Available online at http://www.sfwmd.gov/org/ema/everglades/index.html .[March 26, 2004].

Gilmour, C.C., D.P. Krabbenhoft and W.O. Orem. 2004. Mesocosm Studies to Quantify How Methymercury in the Everglades Responds to Changes in Mercury, Sulfur and Nutrient Loading. Appendix 2B-3. In: 2004 Everglades Consolidated Report. South Florida Water Management District, West Palm Beach, FL. Available online at http://www.sfwmd.gov/org/ema/everglades/index.html [March 26, 2004].

Hancock, J.A., J.A. Kushlan and M.P. Kahl. 1992. *Storks, Ibises and Spoonbills of the World*. Academic Press. London, UK.

Heinz, G.H. 1979. Methylmercury: Reproductive and Behavioral Effects on Three Generations of Mallard Ducks. *J. Wildl. Manage.*, 43: 394-401.

Heinz, G.H. 1996. Mercury Poisoning in Wildlife. In: Fairbrother, A., L.N. Locke, G.L. Hoff, eds. *Noninfectious Diseases of Wildlife.* 2nd edition, pp. 118-127. Iowa State University Press, Ames, IA.

Heinz, G.H., and D.J. Hoffman. 2003. Embryotoxic Thresholds of Mercury: Estimates from Individual Mallard Eggs. *Arch. Environ. Contam. Toxicol.*, 44: 257-264.

Hope, B.K. 2005. Performing spatially and temporally explicit ecological exposure assessments involving multiple stessors. Human and Ecological Risk Assessment, 11: 539-566.

Kahl, M. 1964. Food Ecology of the Wood Stork (*Mycteria americana*) in Florida. *Ecol. Monogr.*, 34: 97-l 17.

Kushlan, J.A. 1978. Feeding Ecology of Wading Birds. A. Sprunt Jr., J.C. Ogden and S.A. Winkler, eds.. In: *Wading Birds.* National Audubon Society, New York, NY.

Kushlan, J.A. 1979. Prey Choice by Tactile-Foraging Wading Birds. *Colonial Waterbirds*, 3: 133-142.

Lange, T., D.A. Richard and H.E. Royals. 1998. Trophic Relationships of Mercury Bioaccumulation in Fish from the Florida Everglades. Report to the Florida Department of Environmental Protection, Florida Game and Fresh Water Fish Commission, Eustis, FL.

Leady B.S., and J.F. Gottgens. 2001. Mercury Accumulation in Sediment Cores and Along Food Chains in Two Regions of the Brazilian Pantanal. *Wetlands Ecology and Management*, 9: 349–361.

Loftus, W.F., J.C. Trexler and R.D. Jones. 1998. Mercury Transfer through an Everglades Aquatic Food Web. Report to Florida Department of Environmental Protection. U.S. Geological Survey – BRD, Everglades N.P. Field Station, Homestead, FL

Nick Miller, Inc. 2003. 2003 Vegetation Map Report: Stormwater Treatment Areas 2, 5, & 6. Report to South Florida Water Management District. Contract No. C-C19905, Work Order 01, West Palm Beach, FL.

Ogden, J.C., J. Kushlan and J.T. Tilmant. 1976. Prey Selectivity by the Wood Stork. *Condor*. 78: 324-330.

Presidential/Congressional Commission on Risk Assessment and Risk Management (1997) *Framework for Environmental Health Risk Management* (Final Report), Vol. 1, Washington DC, USA, Presidential/Congressional Commission on Risk Assessment and Risk Management

Reynolds, J.B. 1996. Electrofishing. B,R. Murphy and D.W. Willis, eds, In: Fisheries Techniques, 2nd ed., pp. 221–254. American Fisheries Society, Bethesda, MD, USA.

Rumbold, D.G. 2000a. Contaminants in Wildlife Inhabiting the Area Around Palm Beach County's North County Resource Recovery Facility: 1989–1999. Final Report to the Solid Waste Authority of Palm Beach County, February 2000. West Palm Beach, FL.

Rumbold, D.G. 2000b. Appendix 7-3b: Methylmercury Risk to Everglades Wading Birds: a Probabilistic Ecological Risk Assessment. G. Redfield, ed. In: *2000 Everglades Consolidated Report*, South Florida Water Management District, West Palm Beach, FL. Online at http://www.sfwmd.gov/org/ema/everglades/index.html [March 26, 2004].

Rumbold D.G. 2004. A Probabilistic Risk Assessment of the Effects of Methylmercury on Great Egrets and Bald Eagles Foraging at Stormwater Treatment Area 2. Report to the Florida Department of Environmental Protection, Tallahassee, FL.

Rumbold D.G. 2005a. A Probabilistic Risk Assessment of the Effects of Methylmercury on Great Egrets and Bald Eagles Foraging at a Constructed Wetland in South Florida Relative to the Everglades. *Human and Ecological Risk Assessment*, 11(2): 365-388.

Rumbold, D.G. 2005b. Appendix 2B-1: Annual Permit Compliance Monitoring Report for Mercury in Downstream Receiving Waters of the Everglades Protection Area. G. Redfield, ed. In: *2005 South Florida Environmental Report – Volume I*. South Florida Water Management District, West Palm Beach, FL. Online at http://www.sfwmd.gov/org/ema/everglades/index.html.

Rumbold, D.G. and L. Fink. 2002. Appendix 4A.6: Report on Expanded Mercury Monitoring at Stormwater Treatment Area 2. In: 2002 Everglades Consolidated Report. South Florida Water Management District, West Palm Beach, FL. Available online at http://www.sfwmd.gov/org/ema/everglades/index.html [March 26, 2004].

Rumbold D.G. and L.E. Fink. In press. Extreme Spatial Variability and Unprecedented Methylmercury Concentrations within a Constructed Wetland. *Environ. Monit. Assessment*.

Ryder, R.A. 1950. Great Blue Heron Killed by a Carp. *Condor*, 52(1): 40-41.

Sample, B. E. and G.W. Suter II. 1999. Ecological Risk Assessment in a Large River-Reservoir: 4. Piscivorous Wildlife. *Environ. Toxicol. Chem.*, 18: 610-630.

Schierow, L.J. 1998. Risk Analysis: Background on Environmental Protection Agency Mandates. CRS Report for Congress. Redistributed by National Library for the Environment.

Spalding, M.G., P.C. Frederick, H.C. McGill, S.N. Bouton, L.J. Richey, I.M. Schumacher, C.G. Blackmore and J. Harrison. 2000. Histologic, Neurologic, and Immunologic Effects of Methylmercury in Captive Great Egrets. *J. Wildl. Disease*, 36: 423-435.

Stalmaster, M.V. And J.A. Gessaman, 1982. Food Consumption and Energy Requirements of Captive Bald Eagles. *J. Wildl. Manage*, 46: 646-654.

Surdick, J.A., Jr. 1998. Biotic and Abiotic Indicators of Foraging Site Selection and Foraging Success of Four Ciconiiform Species in the Freshwater Everglades of Florida. Unpublished Masters Thesis. University of Florida, Gainesville, FL.

Suter, G.W., II. 1993. *Ecological Risk Assessment.* Lewis Publishers, Boca Raton, FL.

Thompson, D.R. 1996. Mercury in Birds and Terrestrial Mammals. W.N. Beyer, G.H. Heinz and A.W. Redmon-Norwood, eds. pp. 341-356. In: *Environmental Contaminants in Wildlife: Interpreting Tissues Concentrations.* Lewis Publications, New York, NY.

USEPA. 1993. Wildlife Exposure Factors Handbook. U.S. Environmental Protection Agency. EPA/600/R-93/187.

USEPA. 1995. Final Water Quality Guidance for the Great Lakes System. U.S. Environmental Protection Agency. 40 CFR Parts 9, 122, 123, 131, and 132.

USEPA. 1997. Mercury Study Report to Congress. Vol. VI: An Ecological Assessment for Anthropogenic Mercury Emissions in the United States. U.S. Environmental Protection Agency. EPA-452/R-97-008.

USEPA. 1998. Guidelines for Ecological Risk Assessment. U.S. Environmental Protection Agency. EPA/630/R-95/002F.

USEPA. 1999. Using Probabilistic Analysis in Ecological Risk Assessment. In: *Risk Assessment Guidance for Superfund Volume 3, Chapter 5*. U.S. Environmental Protection Agency. Part B (DRAFT) EPA 000-0-99-000. Online at http://www.epa.gov/superfund/programs/risk/rags3adt/ [March 26, 2004].

USFWS. 1998. Endangered Species Consultation Handbook: Procedures for Conducting Consultation and Conference Activities under Section 7 of the Endangered Species Act. U.S. Fish and Wildlife Service. March 1998. Online at http://www.fws.gov/endangered/consultations/s7hndbk/s7hndbk.htm [July 18, 2005].

USFWS. 1999. South Florida Multi-Species Recovery Plan. U.S. Fish and Wildlife Service, Atlanta, Georgia.

USFWS. 2003. Evaluation of the Clean Water Act Section 304(a) Human Health Criterion for Methylmercury: Protectiveness for Threatened and Endangered Wildlife in California. U.S. Fish and Wildlife Service, Environmental Contaminants Division, Sacramento, CA. Online at http://sacramento.fws.gov/ec/Methylmercury_20Criterion_20Evaluation_20Final_20Report_20October_202003.pdf [March 26, 2004].

U.S. National Research Council. 1994. Science and Judgment in Risk Assessment. Committee on Risk Assessment of Hazardous Air Pollutants, Board on Environmental Studies and Toxicology, and Commission on Life Sciences, Washington, D.C. National Academy Press.

Welty, J.C. 1982. *The Life of Birds*, 3rd ed., Saunders College Publishing, Philadelphia, PA.

Wiener, J.G., D.P. Krabbenhoft, G.H. Heinz and A.M. Scheuhammer. 2003. Ecotoxicology of Mercury. D.J. Hoffman, B.A. Rattner, G.A. Burton Jr. and J. Cairns Jr., eds. pp. 409-463. In: *Handbook of Ecotoxicology*. Lewis Publishers, Boca Raton, FL.

Windward Environmental LLC. 2003. Lower Duwamish Waterway Remedial Investigation. Appendix A. Phase 1 Ecological Risk Assessment. Final Report to the U.S. Environmental Protection Agency, Region 10, Seattle, WA. Online at http://www.ldwg.org/PubDocArchive.htm [March 26, 2004].

Wolfe, M.F., S. Schwarzbach and R.A. Sulaiman. 1998. Effects of Mercury on Wildlife: A Comprehensive Review. *Environ. Tox. Chem.*, 17: 146-160.

Wood, P.B., M.W. Collopy and C.M. Sekerak. 1998. Postfledging Nest Dependence Period for Bald Eagles in Florida. *J. Wildl. Manage.*, 62: 333-339.